

Original Articles

Use of artificial habitats to detect spawning sites for the conservation of *Galaxias maculatus*, a riparian-spawning fishD.S.E. Orchard^{a,b,*}, M.J.H. Hickford^b, D.R. Schiel^b^a Waterways Centre for Freshwater Management, University of Canterbury, Christchurch, New Zealand^b Marine Ecology Research Group, University of Canterbury, Christchurch, New Zealand

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ABSTRACT

Galaxias maculatus is a diadromous riparian-spawning fish that supports an important fishery. Eggs develop terrestrially as with several other teleost fishes. Spawning habitat occurs in specific locations near rivermouths and its protection is a conservation priority. However, quantifying the areas involved is hampered by high egg mortality rates on degraded waterway margins. We hypothesised that temporary artificial habitat would detect spawning in these situations producing a useful indicator for riparian management. We installed arrays of straw bales as artificial habitat in two independent experiments over consecutive years and assessed their impact using pairwise Before-After-Control-Impact (BACI) experimental designs. We tested degraded gaps within the distribution of known spawning sites and also areas further upstream and downstream. Nine spawning occurrences were recorded on artificial habitats in 2015, 22 in 2016, and two on paired controls. Both experiments produced a significant effect for artificial habitats deployed in degraded gaps within the known spawning site distribution ($p = 0.0001$) providing evidence that these locations should be regarded as actual or potential spawning sites. In 2016 the technique also produced a significant effect downstream of known sites in one of the study catchments ($p = 0.0375$). We believe the use of artificial habitats as a detection tool could be useful in a variety of management contexts. These include identifying areas for protection, as confirmation of site suitability prior to making restoration investments, and in investigations to support the migration of habitats to new locations under climate change, since these may currently be degraded.

1. Introduction

Galaxias maculatus (Jenyns, 1842) is a diadromous fish species that is widely distributed in the Southern Hemisphere (Berra et al., 1996). The harvesting of juveniles during their upstream migration supports lucrative fisheries in several countries (Barbee et al., 2011). However, the species is in decline in New Zealand (Goodman et al., 2014) and South America (Encina-Montoya et al., 2011) prompting concern for the fishery and a range of conservation measures. A major contributing factor is the degradation of spawning habitat associated with land use change in lowland catchments (Hickford et al., 2010). Due to a specialised reproductive strategy the eggs develop in a terrestrial environment (McDowall and Charteris, 2006). This is associated with delayed hatching to coincide with favourable conditions for larval survival (Martin, 1999). Conversely, this increases vulnerability to anthropogenic threats (Hickford and Schiel, 2011a). Other examples of terrestrial egg development in teleost fishes include Mummichog (*Fundulus heteroclitus*), Diamond killifish (*Aidinia xenica*), California grunion (*Leuresthes tenuis*), Gulf grunion (*L. sardine*), and Giant kōkupu

(*Galaxias argenteus*) (Franklin et al., 2015; Martin, 1999). Spawning occurs in riparian vegetation inundated during spring high tides and close to the upstream limit of salt water intrusion (Benzie, 1968). Spatiotemporal variance may result from interactions between salinity, water level, topography and the timing of fish movements and spawning events, making detection of the sites used more difficult (Orchard and Hickford, 2018a). This is a significant issue for management and is usually attempted by direct observations of adult fish during spawning events, or searches of riparian vegetation for eggs (Taylor, 2002). However, both of these approaches have conceptual and practical weaknesses.

For the detection of spawning sites, observations of adult fish have problems with precision unless spawning was actually observed. Adult *G. maculatus* spend several days shoaling in pre-spawning aggregations and devote considerable energy to searching riparian vegetation before selecting a spawning site (Benzie, 1968). There may be a large area in which an aggregation is observed prior to spawning that is relatively imprecise compared to the sites actually used. In comparison, direct searches for eggs provide indisputable evidence that spawning has

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occurred. However, egg mortality between the date of spawning and the field survey reduces the effectiveness of this approach. Recent research has found that spawning may occur irrespective of whether the habitat is favourable for egg survival (Hickford and Schiel, 2011a) and egg mortality can be extremely high (Hickford and Schiel, 2011b). This suggests that egg mortality is a major management issue rather than the absence of spawning *per se*, and the same issue makes the detection of spawning sites more difficult. Once dead, the tiny eggs (approximately 1.2 mm Ø) dehydrate and rapidly disappear (Harzmeyer, 2006). In degraded environments, surveys reliant on egg discovery may fail to detect spawning sites or underestimate the areas involved.

Further research has shown that artificial habitats such as installations of straw bales can provide favourable spawning sites and support high egg survival rates (Hickford and Schiel, 2013). We predicted that temporary installations of artificial habitats could also be used as a detection tool in degraded areas. In particular, we expected that experimental arrays might produce a useful indicator for management to help identify unknown spawning locations or establish the full extent of potential spawning habitat on degraded riparian margins. To test this, we hypothesised that artificial habitats would detect spawning at locations where eggs had not been detected in previous field surveys due to either the influence of egg mortality on survey findings or avoidance of those sites by adult fish, since it is difficult to distinguish directly between the two.

In addition, a test of this hypothesis needed to account for the inability to provide a true control – a conundrum that is typical of before-after experiments (Stewart-Oaten et al., 1986). To address this we assessed the effect of installing straw bales on riverbanks using replicate treatment-control pairs in a modified Before-After-Control-Impact (BACI) experimental design (Underwood, 1992). In this terminology, the experimental approach tests whether an intervention (e.g., the introduction of artificial habitats) has a statistically significant impact on a response variable of interest, such as the occurrence of eggs (Stewart-

Oaten and Murtaugh, 2003). We also considered the application of artificial habitats to two different management questions: whether spawning could be detected at previously unrecorded but currently degraded locations within the distribution of known spawning sites, and whether spawning could also be detected outside of the distribution of known sites where these areas also happened to be degraded. We use the term ‘spawning sites’ to refer to the geospatial position of eggs in the environment. The term ‘spawning habitat’ refers to the locations and physical conditions that support spawning. In this paper our objectives are to i) demonstrate the use of artificial habitats to overcome egg detection issues at degraded locations, and ii) discuss applications of this approach to support conservation planning in the wider management context.

2. Materials and methods

2.1. Study areas and context

The study areas are located in the Avon-Heathcote Estuary/Ihutai catchment in the city of Christchurch on the east coast of New Zealand’s South Island (Fig. 1). This is a barrier-enclosed tidal lagoon system associated with a dynamic sand spit at the southern end of Pegasus Bay (Kirk, 1979). The Avon and Heathcote rivers are spring-fed lowland waterways with average base flows of approximately 2 and 1 m³ s⁻¹ respectively (White et al., 2007). Anzac Creek and an area of interconnected swamps and small lakes are tributaries of the Avon (Fig. 1). The total study area included a reach of 3.5 km in the Heathcote mainstem, 3.5 km in the Avon mainstem, and an additional 0.7 km in the Anzac sub-catchment (Fig. 1).

Although many aspects of the main waterways are similar (White et al., 2007), the Heathcote catchment is modified by a tidal barrage that limits the upstream progression of the tide (Christchurch City Council, 2016). In comparison to the Avon, this reduces salinity in the

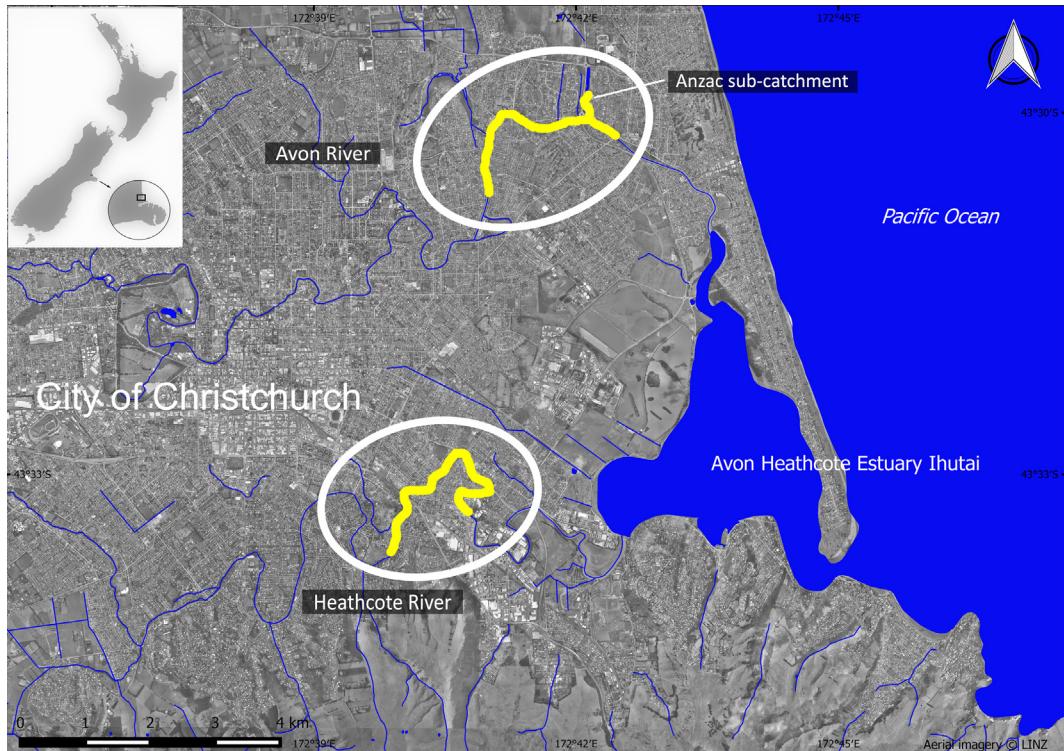


Fig. 1. Location of the city of Christchurch and Avon Heathcote Estuary/Ihutai on the east coast of the South Island, New Zealand, showing the study areas in the Avon and Heathcote River catchments. Yellow lines indicate reaches that were searched for *G. maculatus* spawning sites after the 2010–11 Canterbury earthquakes. The results of these surveys informed the artificial habitat experiment design. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

reach used for spawning. Recent studies have found spawning sites in salinities of up to 18 ppt in the Avon versus only 10 ppt in the Heathcote (Orchard and Measures, 2017). These studies also showed that the downstream limit of spawning was governed by habitat quality characteristics rather than being a direct effect of salinity tolerance, and that upstream limit of spawning coincided with the upstream extent of salt water intrusion in the majority of spawning events (Orchard and Measures, 2017). In other respects the waterway morphologies and riparian vegetation characteristics are similar and these are of particular interest to the technique described here. A considerable proportion of the waterway margins are degraded as a consequence of engineered armouring or vegetation clearance activities of several types (Orchard et al., 2018). Each catchment provides an independent test of the technique which is generally applicable to any tidal river where there is a) a known or suspected fish population, and b) degraded spawning habitat.

The Avon and Heathcote catchments are among New Zealand's most intensively studied areas for *G. maculatus* spawning habitat (Taylor, 2002). However, a sequence of large earthquakes in 2010–2011 caused severe impacts in the east of the city that were pronounced in the vicinity of the estuary (Beavan et al., 2012). Ecological changes were expected at a variety of scales in association with altered hydrodynamics, ground and water levels, and substrates (Measures et al., 2011). This created an imperative for updated information on the distribution of *G. maculatus* spawning sites to inform earthquake recovery planning and longer term waterway management (Orchard and Hickford, 2016; Orchard and Hickford, 2018b). The potential for a large scale shift to have occurred was initially investigated in 2015 using intensive surveys of riparian vegetation (Orchard and Hickford, 2018a). This study expands on that work to address the aforementioned detection issues posed by degraded riparian vegetation.

2.2. Artificial habitats

Following Hickford and Schiel (2013) each artificial habitat installation was a set of straw bales oriented with the longest side perpendicular to the river bank and a 5 cm gap between the faces of adjacent bales. Spawning occurs in the microclimate provided by this gap. Each bale set was secured with a pair of steel stakes between which a loop of wire was fixed and driven down to pin the installation to the riverbank (Fig. 2). The bales were placed to straddle the high waterline on the spring tides of each month (Fig. 3). Water levels were monitored over the spring tide sequences and the bales were repositioned accordingly. In the 2015 experiment, each installation used three bales. In

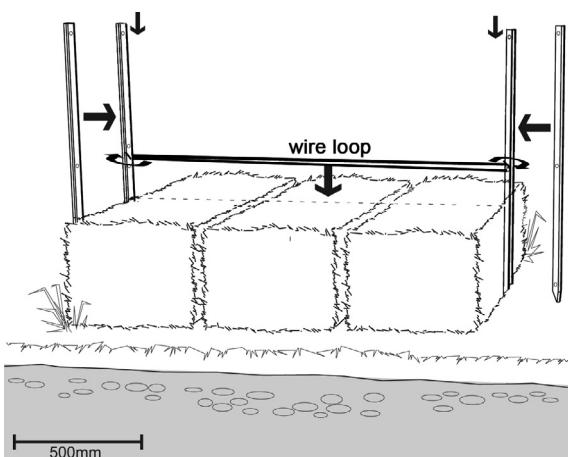


Fig. 2. Design of the artificial spawning habitats using straw bales. Bales are secured by a loop of wire attached to steel stakes which are driven down to pin bales to the ground. The long axis of the bales is oriented perpendicular to the riverbank. Adapted from Hickford and Schiel (2013).



Fig. 3. An artificial habitat installation on a degraded section of riverbank in the Avon catchment in 2015. Photo: S. Orchard.

2016, sets of two bales were used to improve spatial coverage with the resources available. In both years, three replicates were installed at each site and placed at 10–20 m spacing along the bank depending on logistical constraints.

2.3. Experimental design

We conducted two independent experiments in successive years. A pilot study was completed in 2015 using 24 installations followed by a larger-scale experiment ($n = 90$) in 2016 known as the Whaka Inaka – Causing Whitebait project. The design of both experiments was informed by prior field surveys that identified a) habitat quality for *G. maculatus* spawning across the entire study area, and b) the location of natural spawning sites based on the detection of eggs in riparian vegetation.

Habitat quality was assessed by field survey following habitat quality classification schema (Supplementary Material, Table S1) with a minimum classification unit of 5 m riverbank length. Intensive egg surveys were completed for the entire study area over the four peak months of spawning activity in 2015 (Feb–May) and three months in 2016 (Feb–Apr), following Orchard and Hickford (2018a). Each survey commenced six days after the highest tides in the spring tide sequence each month (Supplementary Material, Table S2) and took 10–14 days to complete depending on the number of spawning sites found and weather conditions. In each experiment, three situations of interest for management were identified: degraded locations within the known distribution of naturally occurring spawning sites (i.e., degraded gaps), and both upstream and downstream of the known distributional limits of spawning. In each case ‘degraded’ was defined as areas of Class 1 or 2 (‘poor’ or ‘moderate’) habitat quality (Supplementary Material, Table S1). The distribution of naturally occurring spawning sites was obtained from the results of egg surveys (Orchard et al., 2018). Each spawning site was a continuous or semi-continuous patch of eggs for which the upstream and downstream limits were recorded as point coordinates. In all cases, natural spawning sites did not occur in areas that had been assessed as degraded.

Spatial data for spawning sites and habitat quality were visualised in QGIS v2.18 (QGIS Development Team, 2017). For each waterway where spawning was found (Avon, Heathcote, and Anzac) three experimental reaches were defined according to the portion of the study area i) within, ii) upstream, and iii) downstream, of the known spawning site distribution (Fig. 4, Supplementary Material, Table S3). Throughout this paper these are referred to as the ‘known distribution’, ‘upstream’ and ‘downstream’ experimental reaches. Reach lengths for each were calculated on the centrelines of waterway channels digitised from 0.075 m resolution post-quake aerial photographs (Land Information New Zealand, 2016). Overlay analysis was used to delineate degraded locations with a minimum bank length of 10 m. These locations were the focus of the experimental design.

2.3.1. 2015 Pilot study design

Egg survey results from February and March 2015 were used to delineate the experimental reaches for design of the pilot study, and artificial habitats were installed in April. A ‘test site’ was selected at random within each of the three experimental reaches in the Avon and Heathcote mainstem, and for the ‘known distribution’ and ‘upstream’ reaches in the Anzac sub-catchment resulting in a total of eight test sites (Supplementary Material, Table S3). Following a replicated paired design (Underwood, 1992), three replicate treatment-control pairs were established at the centre point of the degraded locations closest to each test site resulting in a total of 24 artificial habitat installations and their paired controls (Fig. 4a & b). Each replicate comprised of a straw bale set and 1 m of vegetation either side (addressing a halo effect that is sometimes observed) giving a total bank length of 4 m which was considered to be the ‘treatment’, a paired ‘control’ area of the same dimensions, and a 2 m separation between treatment-control pairs. The footprint of each treatment-control pair was monitored in all months (Feb–May). Monthly monitoring took place 7–14 days after the peak spring tide and involved inspection of the stems and root mats of all vegetation in the test areas. For the artificial habitats this included all surfaces of the straw bale sets and the 1 m halo of adjacent vegetation, as described above. Eggs were recorded on a presence-absence basis.

2.3.2. 2016 Whaka Inaka – Causing Whitebait project

In 2016, the Heathcote mainstem and Anzac sub-catchment of the Avon were chosen for a larger scale experiment (Fig. 4). Results from

four months of 2015 egg surveys were used to inform the experimental design. However, this resulted in the same experimental reaches (as used in 2015) being defined in these study areas since the spawning sites found in April and May did not extend the known distribution beyond the sites recorded in February and March (Supplementary Material, Table S3). Compared to the pilot study, a major difference was the installation of artificial habitats in January to facilitate monitoring over the three peak months of spawning activity (February–April) as identified in the 2015 egg surveys (Orchard and Hickford, 2016). The 2016 design also markedly increased sampling effort in each of the two study areas selected with a total of 21 artificial habitats installed in the Anzac sub-catchment and 69 in the Heathcote. A randomised block design (*sensu* Hurlbert, 1984) was used to select the test sites with the blocks being 100 m contiguous reaches within the experimental reaches to be sampled. Due to limitations on materials and monthly monitoring resources only 23 of 35 blocks were sampled in the Heathcote together with seven in the Anzac study area for a total of 30 test sites (Supplementary Material, Table S3). Other details are as per the 2015 experiment except the dimensions of each artificial habitat installation (and its paired control) were smaller due to the use of two straw bales per installation instead of three.

2.4. Data analysis

Our analysis followed a pairwise BACI analytical framework (Stewart-Oaten et al., 1986; Underwood, 1992). Before and after

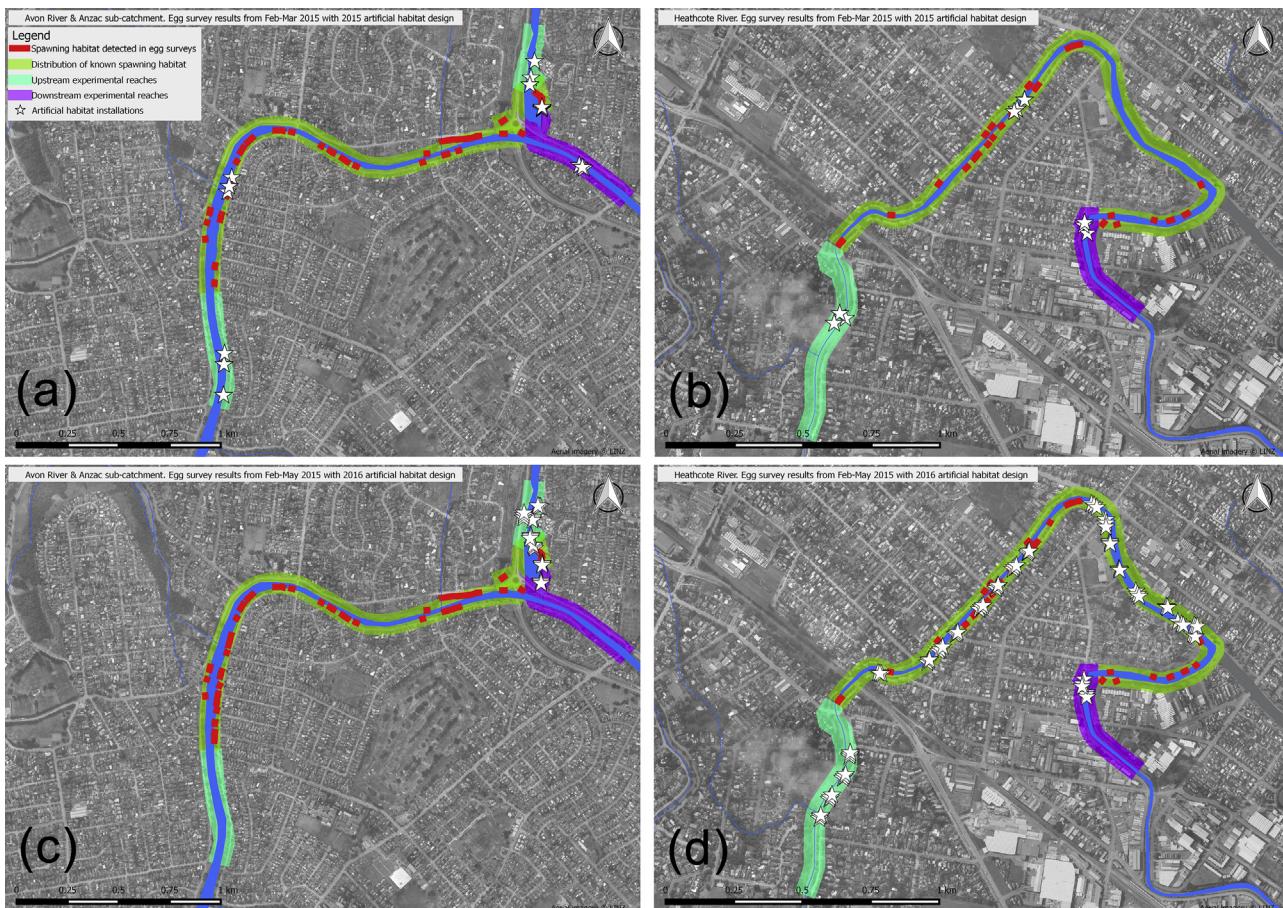


Fig. 4. Location of the experimental reaches as defined by the results of egg surveys completed prior to the installation of artificial habitats. Red lines show the spatial extent of all naturally occurring spawning sites detected in 2015 (n = 85). Coloured areas show the three experimental reaches defined in relation to the location of the confirmed spawning sites. White stars are the locations of individual treatment-control replicates: (a) Avon mainstem and Anzac sub-catchment, 2015, (b) Heathcote mainstem, 2015, (c) Avon mainstem and Anzac sub-catchment, 2016, (d) Heathcote mainstem, 2016. Downstream is to the right in all maps. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

sampling was provided by the results of egg occurrence monitoring at each of the test sites in relation to the study period and timing of the treatment being tested (being the introduction of artificial habitats). Statistical analysis was carried out following a quasi-likelihood generalised linear model method (McDonald et al., 2000) in R v3.3.1 (R Core Team, 2016) using the packages *lme4* (Bates et al. 2015) and *lsmeans* (Lenth, 2016). The statistical model in R notation is:

$$Y = \text{Period} + \text{Treatment} + \text{Period} * \text{Treatment} + (1|\text{Month}) \\ + (1|\text{Catchment}) + (1|\text{Month} * \text{Catchment}) + \text{Period} * \text{Treatment}(R)$$

where *Period* is the classification of time as Before or After, *Treatment* is the classification of sites as either Impact or Control, and their interaction *Period* * *Treatment* represents the BACI contrast of interest. Random effects for month, catchment and their interaction are modelled as $(1|\text{Month})$, $(1|\text{Catchment})$ and $(1|\text{Month} * \text{Catchment})$, and *Period* * *Treatment*(*R*) represents the residual variance.

Analysis was completed on the raw data with the response variable being the count of egg occurrences for each installation together with its control within each Period. All sites were measured in each month of the experiment. The fixed effects are Treatment, Period and their interaction (which is the BACI contrast of interest). The random effects were Catchment and Month. For models specific to one catchment the Catchment term was replaced with Site. Visual inspection of residual plots did not reveal obvious deviations from homoscedasticity or normality. P-values were obtained by likelihood ratio tests in *lsmeans* using the full model containing the effect of interest (*Period* * *Treatment*) against the model with this effect dropped following Bolker et al. (2009).

3. Results

3.1. Spawning occurrences on artificial habitats

Eggs were recorded on artificial habitats in both years (Table 1). Nine occurrences were recorded from a potential maximum of 48 (installation-months) in 2015, and 22 from a potential maximum of 270 in 2016. The highest number of occurrences per month was recorded in March 2016 consistent with the previously reported timing of peak spawning activity (Orchard and Hickford, 2016). Within the known spawning site distribution the highest monthly ‘strike rate’ (egg occurrences per installation) was 0.56 in 2015, and 0.24 in 2016 with means across all installations per year of 0.44 and 0.11 respectively (Table 1). In the upstream experimental reaches only one occurrence was recorded (Anzac sub-catchment, April 2015). None were recorded in any of the upstream experimental reaches in 2016 despite greater

sampling effort. In the downstream experimental reaches no occurrences were recorded in the 2015, but there were five in 2016 (Table 1). These comprised of three in the Heathcote and two in the Anzac study areas. This represented a mean strike rate of 0.19 and was higher than that (0.11) recorded on artificial habitats within the known distribution that year (Table 1).

3.2. BACI analysis

The control site results add important information for the assessment of artificial habitats as an indicator of previously undetected spawning. In both experiments there were no eggs recorded on any of the controls for test sites within the known spawning distribution (Fig. 5). However, there were a total of eight spawning occurrences on artificial habitats in 2015 (Fig. 5a & b), and 18 in 2016 (Fig. 5c & d) in these locations. A single upstream occurrence was recorded in the Anzac sub-catchment in 2015 that was also associated with a negative result on the control (Fig. 5a). In the downstream experimental reaches there were three spawning occurrences recorded in the Heathcote in 2016 and again none on the controls (Fig. 5d). Over the same period in the Anzac sub-catchment two occurrences were recorded on artificial habitats but in both cases there were concurrent occurrences recorded on the paired controls (Fig. 5c).

The BACI analysis confirms the significance of these trends (Table 2). The artificial habitats were found to have a significant effect ($p = 0.0001$) on the detection of egg occurrences in both years when used to test degraded locations from within the known spawning distribution (Table 2a & b). In 2015, the artificial habitats did not produce a significant effect ($p < 0.05$) when used to test degraded locations outside of the known distribution, despite that eggs were recorded on one installation (Table 2a). In 2016, the artificial habitats produced a significant effect in the Heathcote downstream reach ($p = 0.0375$, Table 2b). In the Anzac Creek downstream reach, the artificial habitats did not produce a significant effect despite egg occurrences being recorded (Table 2b). This reflects the control results and was associated with a change in vegetation condition resulting from the recovery of riparian grasses in a previously disturbed area during the summer months.

4. Discussion

4.1. Artificial habitats as a detection tool

Results from both experiments showed that artificial habitats could detect *G. maculatus* spawning sites on degraded riparian margins at locations where spawning had previously not been recorded. In relation

Table 1
Summary of spawning occurrences on artificial habitats.

Experimental reach	n	Spawning occurrences/month		Total	Mean occurrence/month
<i>2015</i>					
		Month 1 (April)	Month 2 (May)		
Known distribution	9	5	3	8	0.44
Upstream	9	1	0	1	0.06
Downstream	6	0	0	0	0
<i>2016</i>					
		Month 1 (February)	Month 2 (March)	Month 3 (April)	
Known distribution	54	0	5	13	18
Upstream	27	0	0	0	0
Downstream	9	0	2	3	5
					0.11
					0.19

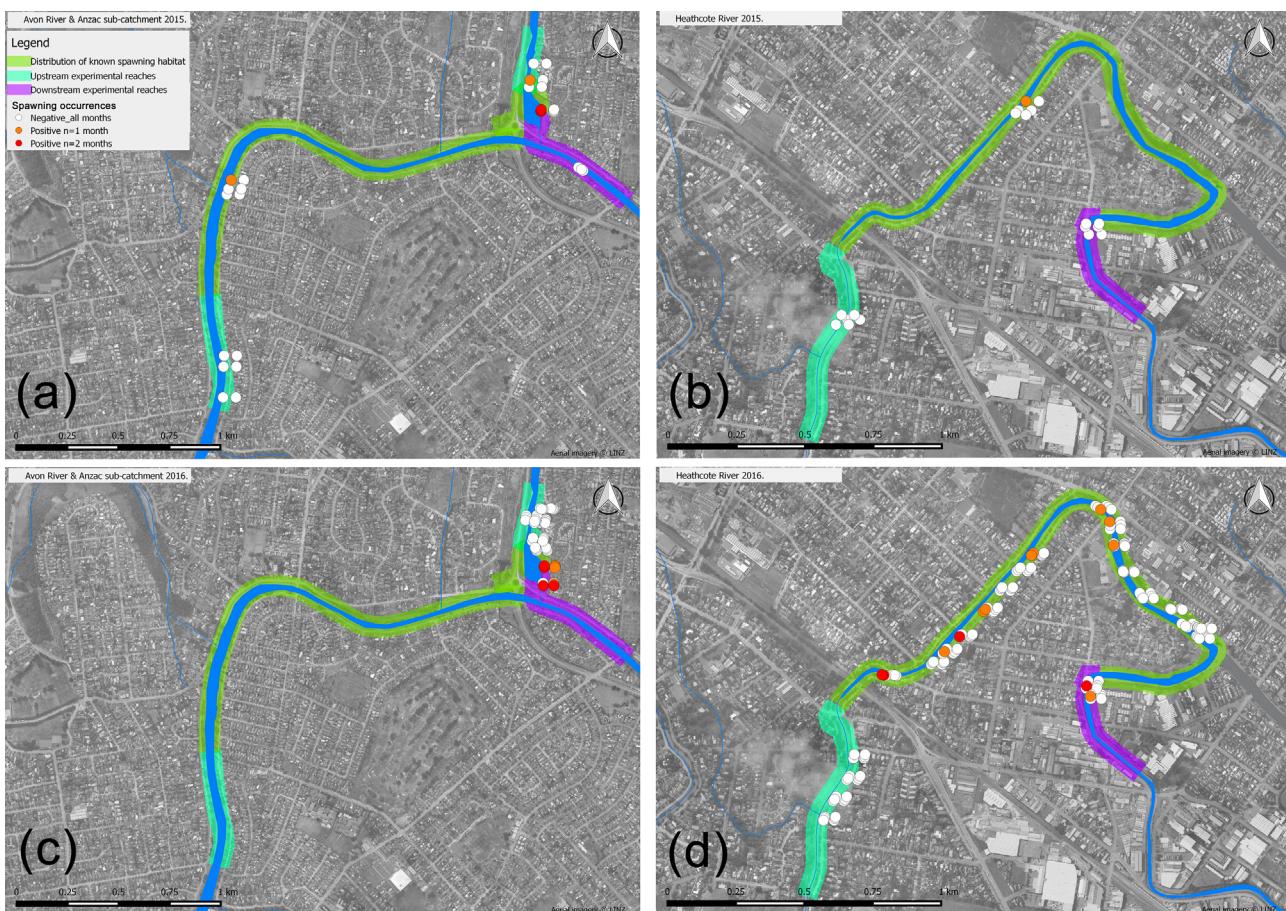


Fig. 5. Number of egg occurrences recorded for each treatment-control pair in the two independent artificial habitat experiments. (a) Avon mainstem and Anzac sub-catchment, 2015, (b) Heathcote mainstem, 2015, (c) Avon mainstem and Anzac sub-catchment, 2016, (d) Heathcote mainstem, 2016. Downstream is to the right in all maps. At each location the left hand dot shows treatment results, the right hand dot is the control.

to our hypothesis, some of these may have been spawning sites that were previously undetected for reasons such as high egg mortality, while others may have been ‘potential’ spawning sites that had previously been avoided by adult fish but became favourable due to the provision of better conditions. Regardless of which of these situations the results reflect, they provide useful information for management in the form of irrefutable evidence that spawning may occur at those locations. Because the physical intervention provided by artificial habitats mimics a structural improvement in the riverbank vegetation (Hickford and Schiel, 2013), sites at which spawning was recorded could be readily improved through ecological restoration techniques.

Our results demonstrate that artificial habitats can be usefully applied as a detection tool to help identify areas for protection or restoration. In many cases, cessation of vegetation clearance and other disturbances is sufficient to address the problem of high egg mortality (Orchard et al., 2018), suggesting that protection is more fundamentally important than active restoration. As such, the technique could help overcome the apparent gap between conservation policy objectives and implementation in practice that has been exacerbated by a lack of spawning site information. In turn, this will contribute to solving the wider conservation issues concerning *G. maculatus* decline and maintenance of a sustainable fishery. These findings extend the literature on the use of artificial habitat methodologies to identify habitat and occupancy patterns characterised by detection challenges. Examples we are aware of include the monitoring of flying (Southwood, 1978) and flightless (Bowie et al., 2014) insects, and cryptic lizard species in forests (Nordberg and Schwarzkopf, 2015) and shrublands (Lettink et al., 2011).

4.2. Do artificial habitats actually improve detection?

In our example the management question concerned the detection of degraded portions of the habitat distribution where spawning was more vulnerable to human pressures yet could be readily restored. This required some form of experimental control to monitor the trajectory of locations identified as ‘degraded’ with and without the introduction of the artificial habitats. Although it is not possible to directly measure whether responses at the treatment site would have remained similar if the treatment not been applied (Bulleri et al., 2008), the control data functions as a body of evidence that assists this estimation and practical interpretation (McDonald et al., 2000). However, if the objective was simply to test for the presence of spawning using artificial habitats as an alternative to direct searches of riparian vegetation, then a simple before-after monitoring design could have been used (Stewart-Oaten and Bence, 2001).

Use of a BACI design allowed us to specifically test whether the artificial habitats had a significant effect on detection (Stewart-Oaten et al., 1986). This was provided by a pairwise sampling strategy with the arrays of artificial habitats treated as the intervention (the hypothetical impact) and analysed as a fixed effect. Although a paired experimental design strengthens the weight of evidence (Underwood, 1994), the inference gained remains subject to the assumption that the trajectories of responses will be exactly parallel between treatment and control units (Murtaugh, 2002). The choice of control is also critical to address the counterfactual of interest (Stewart-Oaten, 2008) and may include asymmetric designs, for example with a greater number of control units than treatments (Underwood, 1994). Although in our case

Table 2
Selected BACI analysis results. Means are those predicted by the model for the impact + after (artificial habitat) versus control + after (control site) comparisons. The BACI estimate (also known as the 'BACI contrast') is the modelled difference in the means attributable to the treatment (being introduction of the artificial habitats). The χ^2 statistic is calculated on the likelihood ratio test for fitted vs. null models.

(a) 2015 Pilot study											
Catchment	Experimental Reach	Model [†]	Ismean ^{ImpactAfter}	SE	Ismean ^{ControlAfter}	SE	χ^2	p (> F) [#]	BACI estimate [‡]	Lower CI	Upper CI
All	Known distribution	Period + Treatment + Period * Treatment + (1 Month) + - (1 Catchment) + (1 Month * Catchment)	0.444	0.088	0.000	0.088	16.096	0.0001	0.444	0.234	0.655
All	Upstream + Downstream	Period + Treatment + Period * Treatment + (1 Month) + - (1 Catchment) + (1 Month * Catchment)	0.038	0.020	0.004	0.020	1.036	0.3087	0.033	-0.031	0.098
All	Upstream	Period + Treatment + Period * Treatment + (1 Month) + - (1 Catchment) + (1 Month * Catchment)	0.056	0.028	0.000	0.028	1.051	0.3053	0.056	-0.056	0.167
Anzac	Upstream (Data = one month After)	Period + Treatment + Period * Treatment + (1 Month) + - (1 Site) + (1 Month * Site)	0.333	0.126	0.000	0.126	2.775	0.0958	0.333	-0.138	0.805
Anzac	Upstream (Data = two months 'After')	Period + Treatment + Period * Treatment + (1 Month) + - (1 Site) + (1 Month * Site)	0.167	0.083	0.000	0.083	1.171	0.2792	0.167	-0.183	0.517
(b) 2016 Whaka Inaka – Causing Whitebait project		Model [†]	Ismean ^{ImpactAfter}	SE	Ismean ^{ControlAfter}	SE	χ^2	p (> F) [#]	BACI estimate [‡]	Lower CI	Upper CI
All	Known distribution	Period + Treatment + Period * Treatment + (1 Month) + - (1 Catchment) + (1 Month * Catchment)	0.178	0.087	0.079	0.087	14.879	0.0001	0.099	0.049	0.149
All	Upstream + Downstream	Period + Treatment + Period * Treatment + (1 Month) + - (1 Catchment) + (1 Month * Catchment)	0.046	0.015	0.019	0.015	1.421	0.2333	0.028	-0.018	0.074
All	Downstream	Period + Treatment + Period * Treatment + (1 Month) + - (1 Catchment) + (1 Month * Catchment)	0.185	0.061	0.074	0.061	1.593	0.2069	0.111	-0.065	0.287
Heathcote	Downstream	Period + Treatment + Period * Treatment + (1 Month) + - (1 Site) + (1 Month * Site)	0.074	0.019	0.000	0.019	4.330	0.0375	0.074	0.004	0.144
Anzac	Downstream	Period + Treatment + Period * Treatment + (1 Month)	0.222	0.128	0.333	0.128	0.299	0.5848	-0.111	-0.541	0.319

[†] *InterO* model fit in R notation.

[#] bold = significant at $p < 0.05$.

[‡] estimated BACI contrast from model.

the design was symmetrical, the experimental controls produced useful information for management. For example, in the Anzac downstream reach in 2016, the control results showed that riparian conditions did not remain degraded. Unanticipated recovery of vegetation was observed to occur over the month of March leading to a situation where a previously degraded location had recovered and spawning was detected.

4.3. Limitations in relation to experimental design

In the selected BACI models presented (Table 2) the analysis showed three independent situations where the artificial habitat technique significantly increased the detection of eggs. These were in degraded gaps within the known distribution in both years for the pooled data across all study areas (Table 2a & b), and downstream of the known distribution in the Heathcote in 2016 (Table 2b). It is also important to note that the strike rate of positive occurrences was highly variable in space and time across the study area. This was likely influenced by the location of test sites in relation to the distribution of the adult fish populations on spawning tides. However, there were few instances where more than one of the three replicates per site received concurrent positive occurrences. Together, these results suggest that it would be unwise to rely on a small number of test sites as a detection measure and that replication was important.

Other limitations of our technique, with a bearing on detectability, included the positioning of artificial habitats in relation to the waterline during spawning tides and potentially other nuances affecting their discoverability by adult fish, as well as observer error in the form of false negatives during monitoring. A further complicating factor could arise from flood events experienced within the monitoring period since larval development can occur in as little two weeks (Benzie, 1968) and hatching will occur if the eggs are inundated after that time (Benzie, 1968). The same situation could arise in regions where subsequent spring tides are of similar amplitude. As a result, hatching (and in some cases spawning) can occur on a two weekly cycle (Taylor, 2002) and a monitoring regime to match that period would be required. Fortunately, the pattern of spring-neap tidal variability in our study area is characterised by one tidal sequence of greater amplitude per month (Walters et al., 2001). This is associated with a monthly spawning event and allowed ample time for monitoring. In the case of *G. maculatus*, the relationship between spawning and tidal cycles also simplifies decisions on sampling frequency. For applying the technique to other species and life stages, appropriate means of addressing temporal variance are likely to take different forms.

4.4. Application to management questions

Accounting for spatiotemporal variance in the detectability of response variables is a particular concern for monitoring, and BACI designs are no exception (Russell et al., 2015). The novel use of artificial habitats as a detection tool offers a benefit in our case due to the installations supporting very high egg survival rates. This goes some way towards overcoming the problem that may arise if effects such as gradients act unevenly on experimental units over time (Hurlbert, 1984) and the data collection process cannot be simultaneous. This also offers a practical advantage in providing flexibility for the researcher as to the exact timing of the field measurements.

Temporal aspects of the specific management question may also be of interest. An example is found in our pilot study where a single positive occurrence was recorded in the Anzac upstream reach at the Anzac site. Treated in isolation, this observation provided irrefutable evidence that spawning could occur in a degraded area beyond the known extent of the habitat distribution. Although BACI analysis may be applied to sub-units of a larger study design (Underwood and Chapman, 2003), inferences drawn from a single month and small number of sites may be of limited value. For example, the result might

reflect a stochastic effect such as a salinity spike that happened to occur in that month, but which is not otherwise typical. Consequently, we applied the analysis to results from all months of the pilot study since this was a better match for the management question of interest. No positive occurrences were recorded in the following month and the overall BACI effect was small and not significant at $p < 0.05$. The same area was sampled using the technique in 2016 and again no positive occurrences were recorded.

The larger scale 2016 experiment was specifically geared towards key questions for management. In this case a randomised block design was used to guide the allocation of sampling effort across a relatively extensive study area. This required a large experimental design to cover the degraded areas of interest, but results were immediately useful for management. For example, in addition to several degraded gaps that were identified as actual or potential spawning sites there was a contiguous reach in the lower Heathcote where the artificial habitats failed to detect egg occurrences (Fig. 5d). This is a neglected part of the river where competing demands on riparian management are considerably less than in other locations creating a favourable setting for restoration works. However, the provision of isolated patches of favourable habitat did not attract spawning there despite lying within the known spawning site distribution. This suggests it would be unwise to attempt habitat restoration in this area until the effect can be explained. Adult fish may avoid this area which is mostly shaded by tall exotic trees. If enhancement for *G. maculatus* is desired, the best strategy may involve extending the areas of currently favourable habitat at either end of this reach until the potential behavioural or discoverability issues are better known.

We believe the use of artificial habitats as a detection tool could be useful in a variety of management contexts. These include identifying areas for protection to address conservation objectives where detectability is problematic. As demonstrated in this case, spawning occurrences in artificial habitats may influence the results of investigations for the detection of spawning sites. Despite the absence of previous spawning observations, these degraded locations may then be regarded as spawning habitat and this attracts statutory protection under national legislation. Identifying degraded habitat that could be readily improved is a related and practical application. In our case the areas concerned were degraded by anthropogenic activities and would likely recover naturally if these disturbances were controlled. In other contexts where active restoration is required, artificial habitat deployment could be a useful means to confirm site suitability prior to making restoration investments. Investigations using this technique are also well suited to supporting the migration of critical habitats to new locations where factors such as climate change are driving range contraction (Hovick et al., 2016). This is a topic of increasing need (Burrows et al., 2014) that requires compensatory habitat expansion into areas that may currently be degraded.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecolind.2018.03.061>.

References

- Barbee, N.C., Hale, R., Morrongiello, J., Hicks, A., Semmens, D., Downes, B.J., Swearer, S.E., 2011. Large-scale variation in life history traits of the widespread diadromous fish, *Galaxias maculatus*, reflects geographic differences in local environmental conditions. Mar. Freshwater Res. 62 (7), 790–800. <http://dx.doi.org/10.1071/mf10284>.
- Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Statist. Software 67 (1), 1–48. <http://dx.doi.org/10.18637/jss.v067.i01>.
- Beavan, J., Motagh, M., Fielding, E.J., Donnelly, N., Collett, D., 2012. Fault slip models of the 2010–2011 Canterbury, New Zealand, earthquakes from geodetic data and observations of postseismic ground deformation. N. Z. J. Geol. Geophys. 55 (3), 207–221. <http://dx.doi.org/10.1080/00288306.2012.697472>.
- Benzie, V., 1968. Some ecological aspects of the spawning behaviour and early development of the common whitebait *Galaxias maculatus attenuatus* (Jenyns). Proc. N. Z. Ecol. Soc. 15, 31–39.
- Berra, T.M., Crowley, L., Ivantsoff, W., Fuerst, P.A., 1996. *Galaxias maculatus*: an explanation of its biogeography. Mar. Freshwater Res. 47 (6), 845. <http://dx.doi.org/10.1071/MF9960845>.
- Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H., White, J.-S.S., 2009. Generalized linear mixed models: a practical guide for ecology and evolution. Trends Ecol. Evol. 24 (3), 127–135. <http://dx.doi.org/10.1016/j.tree.2008.10.008>.
- Bowie, M.H., Allen, W.J., McCaw, J., Heugten, R.V., 2014. Factors influencing occupancy of modified artificial refuges for monitoring the range-restricted Banks Peninsula tree weta *Hemideina ricta* (Anostostomatidae). N. Z. J. Ecol. 38 (1), 132–138.
- Bulleri, F., Underwood, A.J., Benedetti-Cecchi, L., 2008. The analysis of ecological impacts in human-dominated environments: reply to Stewart-Oaten (2008). Environ. Conserv. 35 (1), 11–13. <http://dx.doi.org/10.1017/S037689290800458X>.
- Burrows, M.T., Schoeman, D.S., Richardson, A.J., Molinos, J.G., Hoffmann, A., Buckley, L.B., Poloczanska, E.S., 2014. Geographical limits to species-range shifts are suggested by climate velocity. Nature 507 (7493), 492–495. <http://dx.doi.org/10.1038/nature12976>.
- Christchurch City Council, 2016. Ōpāwaho/Heathcote River Catchment. Tauāki Wai Pātua/Vision and Values. Christchurch City Council & Opus International Consultants, pp. 87.
- Encina-Montoya, F., Vega-Aguayo, R., Mardones-Lazcano, A., Rueda, T., Tello, A., 2011. Characterization of whitebait (*Galaxias maculatus*) respiratory rates to optimize intensive culture carrying capacities. Aquacult. Res. 42 (6), 835–843. <http://dx.doi.org/10.1111/j.1365-2109.2010.02692.x>.
- Franklin, P.A., Smith, J., Baker, C.F., Bartels, B., Reeve, K., 2015. First observations on the timing and location of giant kōkopū (*Galaxias argenteus*) spawning. N. Z. J. Mar. Freshwater Res. 49 (3), 419–426. <http://dx.doi.org/10.1080/00288330.2015.1045004>.
- Goodman, J.M., Dunn, N.R., Ravenscroft, P.J., Allibone, R.M., Boubée, J.A.T., David, B.O., et al., 2014. Conservation status of New Zealand freshwater fish, 2013. New Zealand Threat Classification Series. Department of Conservation, Wellington, pp. 12.
- Harzmeyer, J.R., 2006. Effects of Salinity and Temperature on Hatchability, and Early Development of *Galaxias maculatus* (Honours Dissertation). Deakin University, Victoria.
- Hickford, M.J.H., Cagnon, M., Schiel, D.R., 2010. Predation, vegetation and habitat-specific survival of terrestrial eggs of a diadromous fish, *Galaxias maculatus* (Jenyns, 1842). J. Exp. Mar. Biol. Ecol. 385 (1–2), 66–72. <http://dx.doi.org/10.1016/j.jembe.2010.01.010>.
- Hickford, M.J.H., Schiel, D.R., 2011a. Population sinks resulting from degraded habitats of an obligate life-history pathway. Oecologia 166 (1), 131–140. <http://dx.doi.org/10.1007/s00442-010-1834-7>.
- Hickford, M.J.H., Schiel, D.R., 2011b. Synergistic interactions within disturbed habitats between temperature, relative humidity and UVB radiation on egg survival in a diadromous fish. PLoS One 6 (9), e24318. <http://dx.doi.org/10.1371/journal.pone.0024318>.
- Hickford, M.J.H., Schiel, D.R., 2013. Artificial spawning habitats improve egg production of a declining diadromous fish, *Galaxias maculatus* (Jenyns, 1842). Restor. Ecol. 21 (6), 686–694. <http://dx.doi.org/10.1111/rec.2008.012008>.
- Hovick, T.J., Allred, B.W., McGranahan, D.A., Palmer, M.W., Dwayne Elmore, R., Fuhlendorf, S.D., 2016. Informing conservation by identifying range shift patterns across breeding habitats and migration strategies. Biodivers. Conserv. 25 (2), 345–356. <http://dx.doi.org/10.1007/s10531-016-1053-6>.
- Hurlbert, S.H., 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Monogr. 54 (2), 187–211. <http://dx.doi.org/10.2307/1942661>.
- Kirk, R.M., 1979. Dynamics and Management of Sand Beaches in Southern Pegasus Bay. Morris and Wilson Consulting Engineers Limited, Christchurch.
- Land Information New Zealand, 2016. Christchurch 0.075 m Urban Aerial Photos (2015–16). Imported on Oct. 3, 2016 from 6993 GeoTIFF Sources in NZGD2000/New Zealand Transverse Mercator 2000.
- Lenth, R.V., 2016. Least-squares means: the R package lsmeans. J. Stat. Software 69 (1), 1–33. <http://dx.doi.org/10.18637/jss.v069.i01>.
- Lettink, M., O'Donnell, C.F.J., Hoare, J.M., 2011. Accuracy and precision of skink counts from artificial retreats. N. Z. J. Ecol. 35 (3), 236–246.
- Martin, K.L.M., 1999. Ready and waiting: delayed hatching and extended incubation of anamniotic vertebrate terrestrial eggs. Am. Zool. 39 (2), 279–288.
- McDonald, T.L., Erickson, W.P., McDonald, L.L., 2000. Analysis of count data from before-after control-impact studies. J. Agric. Biol. Environ. Stat. 5 (3), 262–279.
- McDowall, R.M., Charteris, S.C., 2006. The possible adaptive advantages of terrestrial egg deposition in some pelagic diadromous galaxiid fishes (Teleostei: Galaxiidae). Fish Fish. 7 (3), 153–164. <http://dx.doi.org/10.1111/j.1467-2979.2006.00217.x>.
- Measures, R., Hicks, M.D., Shankar, U., Bind, J., Arnold, J., Zeldis, J., 2011. Mapping Earthquake Induced Topographical Change and Liquefaction in The Avon-Heathcote Estuary. Environment Canterbury Report No. U11/13. Environment Canterbury, Christchurch, pp. 28.
- Murtaugh, P.A., 2002. On rejection rates of paired intervention analysis. Ecology 83 (6), 1752–1761. [http://dx.doi.org/10.1890/0012-9658\(2002\)083\[1752:ORROPI\]2.0.CO;2](http://dx.doi.org/10.1890/0012-9658(2002)083[1752:ORROPI]2.0.CO;2).
- Nordberg, E.J., Schwarzkopf, L., 2015. Arboreal cover boards: using artificial bark to sample cryptic arboreal lizards. Herpetologica 71 (4), 268–273. <http://dx.doi.org/10.1653/HERPETOLOGICA-D-15-00016>.
- Orchard, S., Hickford, M., 2016. Spatial Effects of the Canterbury Earthquakes on īnanga Spawning Habitat and Implications for Waterways Management. Report prepared for IPENZ Rivers Group and Ngāi Tahu Research Centre. Waterways Centre for Freshwater Management and Marine Ecology Research Group. University of Canterbury, Christchurch, pp. 37.
- Orchard, S., Hickford, M.J.H., 2018. Census survey approach to quantifying īnanga spawning habitat for conservation and management. N. Z. J. Mar. Freshwater Res. 52 (2), 284–294. <http://dx.doi.org/10.1080/00288330.2017.1392990>.
- Orchard, S., Hickford, M.J.H., 2018b. Protected area effectiveness for fish spawning habitat in relation to earthquake-induced landscape change. bioRxiv 283333. <http://dx.doi.org/10.1101/283333>.
- Orchard, S., Hickford, M.J.H., Schiel, D.R., 2018. Earthquake-induced habitat migration in a riparian spawning fish has implications for conservation management. Aquat. Conserv. Mar. Freshwater Ecosyst. 2018, 1–11. <http://dx.doi.org/10.1002/aqc.2898>.
- Orchard, S., Measures, R., 2017. In: Sea Level Rise Impacts in the Avon Heathcote Estuary Ihutai. Salinity Intrusion and īnanga Spawning Scenarios Report prepared for Christchurch City Council, pp. 56.
- QGIS Development Team, 2017. QGIS Geographic Information System. Open Source Geospatial Foundation Project. <http://qgis.org>.
- R Core Team., 2016. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
- Russell, J.C., Stjernman, M., Lindström, Å., Smith, H.G., Biologiska, I., Centre for E., et al., 2015. Community occupancy before-after-control-impact (CO-BACI) analysis of Hurricane Gudrun on Swedish forest birds. Ecol. Appl. 25 (3), 685–694. <http://dx.doi.org/10.1080/14-0645.1>.
- Southwood, R.S., 1978. Ecological Methods: With Particular Reference to the Study of Insect Populations, 2d, rev. ed. Chapman and Hall, New York; London.
- Stewart-Oaten, A., 2008. Chance and randomness in design versus model-based approaches to impact assessment: comments on Bulleri et al. (2007). Environ. Conserv. 35 (1), 8–10. <http://dx.doi.org/10.1017/S0376892908004566>.
- Stewart-Oaten, A., Bence, J.R., 2001. Temporal and spatial variation in environmental impact assessment. Ecol. Monogr. 71 (2), 305–339. [http://dx.doi.org/10.1890/0012-9615\(2001\)071\[305:TSAVIE\]2.0.CO;2](http://dx.doi.org/10.1890/0012-9615(2001)071[305:TSAVIE]2.0.CO;2).
- Stewart-Oaten, A., Murdoch, W.W., Parker, K.R., 1986. Environmental impact assessment: “pseudoreplication” in time? Ecology 67 (4), 929–940. <http://dx.doi.org/10.2307/1939815>.
- Stewart-Oaten, A., Murtaugh, P.A., 2003. On rejection rates of paired intervention analysis: comment/reply. Ecology 84 (10), 2795.
- Taylor, M.J., 2002. The National īnanga Spawning Database: Trends and Implications for Spawning Site Management. Science for Conservation 188. Department of Conservation, Wellington, pp. 37.
- Underwood, A.J., 1992. Beyond BACI: the detection of environmental impacts on populations in the real, but variable, world. J. Exp. Mar. Biol. Ecol. 161 (2), 145–178. [http://dx.doi.org/10.1016/0022-0981\(92\)90094-Q](http://dx.doi.org/10.1016/0022-0981(92)90094-Q).
- Underwood, A.J., 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. Ecol. Appl. 4 (1), 4–15.
- Underwood, A.J., Chapman, M.G., 2003. Power, precaution, Type II error and sampling design in assessment of environmental impacts. J. Exp. Mar. Biol. Ecol. 296 (1), 49–70. [http://dx.doi.org/10.1016/S0022-0981\(03\)00304-6](http://dx.doi.org/10.1016/S0022-0981(03)00304-6).
- Walters, R.A., Goring, D.G., Bell, R.G., 2001. Ocean tides around New Zealand. N. Z. J. Mar. Freshwater Res. 35 (3), 567–579. <http://dx.doi.org/10.1080/00288330.2001.9517023>.
- White, P.A., Goodrich, K., Cave, S., Minni, G., 2007. Waterways, Swamps and Vegetation of Christchurch in 1856 and Baseflow Discharge in Christchurch City Streams. GNS Science Consultancy Report 2007/103. Taupo.